Water Quality Impacts of Converting to a Poultry Litter Fertilization Strategy

R. D. Harmel,* H. A. Torbert, B. E. Haggard, R. Haney, and M. Dozier

ABSTRACT

When improperly managed, land application of animal manures can harm the environment; however, limited watershed-scale runoff water quality data are available to research and address this issue. The water quality impacts of conversion to poultry litter fertilization on cultivated and pasture watersheds in the Texas Blackland Prairie were evaluated in this three-year study. Edge-of-field N and P concentrations and loads in surface runoff from new litter application sites were compared with losses under inorganic fertilization. The impact on downstream nutrient loss was also examined. In the fallow year with no fertilizer application, nutrient losses averaged 3 kg N ha^{-1} and 0.9 kg P ha^{-1} for the cultivated watersheds and were below 0.1 kg ha^{-1} for the pasture watersheds. Following litter application, PO_4–P concentrations in runoff were positively correlated to litter application rate and Mehlich-3 soil P levels. Following litter application, NO_3–N and NH_4–N concentrations in runoff were typically greater from cultivated watersheds, but PO_4–P concentrations were greater for the pasture watersheds. Total N and P loads from the pasture watersheds (0.2 kg N ha^{-1} and 0.7 kg P ha^{-1}) were significantly lower than from the cultivated watersheds (32 kg N ha^{-1} and 5 kg P ha^{-1}) partly due to lower runoff volumes from the pasture watersheds. Downstream N and P concentrations and per-area loads were much lower than from edge-of-field watersheds. Results demonstrate that a properly managed annual litter application (4.5 Mg ha^{-1} or less depending on litter N and P content) with supplemental N should supply necessary nutrients without detrimental water quality impacts.

As a result of the shift to fewer and larger confined animal operations, environmental and economic issues associated with utilization or disposal of animal manures and litters has become a focal point of conservation efforts (Ribaudo et al., 2003; USDA and USEPA, 1999). Manure has been viewed at times as a waste product requiring disposal, but factors such as increased commercial fertilizer costs and ecological problems associated with mismanaged manure disposal have created a shift in attitude to view animal manures as potential resources that must be managed with care (Janzen et al., 1999; Leidner, 2002). Pending water quality legislation and regulation may also force accelerated change in animal waste management. The problem with utilizing the soil amendment value in manures arises because the applied nutrients often exceed the agronomic needs of the nearby lands. These regional imbalances have resulted from the increase in the number of large confined animal operations and the separation of animal and feed production operations (Ribaudo et al., 2003; USDA and USEPA, 1999). However, when properly managed, land-applied manures provide valuable nutrients and organic matter to agricultural production systems without creating harm to the environment or public health (Sharpley et al., 1994; Janzen et al., 1999). A joint USDA–USEPA report (USDA and USEPA, 1999) states that "land application is the most common, and usually most desirable method of utilizing manure because of the nutrients and organic matter." Although this is a commonly accepted viewpoint, the impact of land-applied animal manures on environmental quality is not well understood.

Field- and watershed-scale runoff water quality data are limited, especially for poultry litter application areas, because of collection difficulties caused by natural rainfall variation, substantial land area needs, and field personnel or automated sampling equipment requirements (Gilley and Risse, 2000; Harmel et al., 2003). However, a few researchers have made the significant commitment to monitor field-scale surface water quality from poultry litter application fields. Site information and load results for these studies, which were conducted on small field plots in pasture and cultivated fields, appear in Table 1. They reported a wide range of nutrient concentration and loads, thus supporting the need for additional long-term studies to explore the factors responsible for differing results.

Recent studies, such as Gburek and Sharpley (1998), Pionke et al. (1999), Haggard et al. (2003), and Green and Haggard (2001), have also quantified nutrient levels in runoff water for larger mixed land use watersheds receiving animal manures and inorganic fertilizers. Gburek and Sharpley (1998) used selected storm events monitored on a 7.3-km^2 watershed in Pennsylvania to explore the impact of critical source areas on P transport and concluded that management of P export should focus on soil P levels in areas likely to produce runoff. Pionke et al. (1999) used historical data from the same watershed to determine seasonal differences in nutrient transport and illustrated the dominance of storm flow in dissolved P export. Based on data collected from four watersheds in northwestern Arkansas, Haggard et al. (2003) recognized the importance of land use on nutrient transport and the potential water quality impact of even small losses of land-applied nutrients. Green and Haggard (2001) used data collected from the Illinois River in northwestern Arkansas to explore the occurrence of and differences between the contribution of point and nonpoint source fate and transport of nutrients. Studies at both scales, large watersheds and edge-of-field, are needed to determine the impact of individual land-application sites and the integrated impact of

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Abbreviations: EMC, event mean concentration; UAN, urea ammonium nitrate.

Table 1. Field-scale water quality studies on poultry litter application sites.

Site	Land use	Dates	Area	Poultry fertilizer	Applied TN	Applied TP	Loss in surface runoff NO_3–N	NH_4–N	TN	PO_4–P	TP
			ha		mean annual kg ha^{-1}						
Iowa†	corn	1998–2000	0.40	manure	150	267	2.5	NA‡	NA	0.2	NA
					301	441	2.8	NA	NA	0.3	NA
Arkansas§	fescue (grazed)	1992–1994	1.23	manure	392	164	0.27	0.40	5.58	4.34	NA
			1.06	litter	357	118	0.28	0.99	3.91	1.58	NA
Alabama¶	corn–winter rye	1991–1993	0.11	litter	312	220	2.66	1.30	5.89	0.48	0.99
					624	440	6.32	2.87	12.69	1.47	2.42
Georgia#	fescue	1995–1996	0.75	litter	638	236	NA	7.50	NA	7.40	NA
Arkansas††	cotton	1996–1998	0.60	litter	240	112	2.2	0.2	4.5	0.4	3.0
Georgia‡‡	bermuda–fescue	1995–1996	0.45	litter	283	113	NA	NA	NA	NA	0.1
					535	236	NA	NA	NA	NA	0.4

† Chinkuyu et al. (2002).
‡ Data not reported in this study.
§ Edwards et al. (1996).
¶ Hall (1994) and Wood et al. (1999).
# Pierson et al. (2001).
†† Vories et al. (2001).
‡‡ Vervoort et al. (1998).

land-application sites and other nonagricultural sources of nutrients, such as home lawns, wastewater treatment plants, and natural sources.

Many small plot studies with simulated and natural rainfall have been conducted to evaluate the impacts of land-applied poultry litter on runoff water quality (e.g., Edwards et al., 1995; Sauer et al., 2000; Nichols et al., 1994) and other manures (e.g., Torbert et al., 2002; Kleinman et al., 2002; Bundy et al., 2001; Eghball et al., 2002). These controlled, replicated experiments are excellent investigative tools, and their results established the current understanding of nutrient loss mechanisms occurring at the point of manure application. Plot studies are not, however, designed to relate to landscape and watershed process. Therefore, based on the need for field- and small watershed–scale water quality data, this study was designed to evaluate the surface water quality impacts of poultry litter applied to cultivated and pasture areas, which is important to the growing Texas poultry industry (USDA and United States Department of Commerce, 1996; USDA National Agricultural Statistics Service, 1999). The major objective was to evaluate edge-of-field nutrient losses in surface runoff resulting from converting to a poultry litter fertilization strategy by comparison with control watersheds with inorganic fertilizer management. This evaluation focused on the hypothesis that increasing litter application rate will lead to increased nutrient concentrations and loads. The collected data were also examined to determine possible relationships between water quality and land use of application fields, fertilizer type on cultivated fields, and litter application on larger downstream watersheds with mixed land use.

MATERIALS AND METHODS

To evaluate the impact of litter application rate on edge-of-field water quality, poultry litter was applied to six cultivated watersheds at rates of 0.0, 4.5, 6.7, 9.0, 11.2, and 13.4 Mg ha^{-1} and to four pasture watersheds at rates of 0.0, 0.0 (grazed), 6.7, and 13.4 Mg ha^{-1}. Water quality was measured at these edge-of-field watersheds, which are homogeneous land use fields and true watersheds, and at three downstream sites with heterogeneous land use to evaluate scale effects. To date, three years of data have been collected. The first year (August 2000 through July 2001) represented fallow conditions with no fertilizer applied. In the second year (August 2001 through July 2002), the initial annual litter application was made, and application was repeated in the third year (August 2002 through July 2003).

Watershed Management

The 10 edge-of-field watersheds and three downstream watersheds selected for this study are located at the USDA-ARS Grassland, Soil and Water Research Laboratory near Riesel, TX (Table 2, Fig. 1). The research site is dominated by Houston Black clay soils (fine, smectitic, thermic Udic Haplusterts). These soils are classic Vertisols and thus shrink and swell considerably as moisture content changes. Litter application rates from 0.0 to 13.4 Mg ha^{-1} were determined a priori and then randomly assigned to each of the six cultivated watersheds (Table 2). Watershed Y6 served as the control for the cultivated watersheds and as such received only inorganic fertilizer. The range of litter rates was chosen to encompass and exceed the entire range of expected application rates used by farmers. Two specific pasture watersheds (SW12 and SW17) were determined to receive no litter and serve as controls because of management as a native prairie and a grazed pasture. Litter rates for the other two pasture watersheds were determined a priori and then randomly assigned (Table 2). The cultivated watersheds were scheduled to receive poultry litter before corn planting in the spring of 2001. However, the combination of unusually wet conditions (Fig. 2) and cool temperatures in the fall and winter of 2000–2001 kept soils from drying and prevented fertilizer application (typically applied in January or February) and corn planting (typically planted from mid-February through March). Because the cultivated watersheds were too wet to fertilize or plant in 2001, this presented an opportunity to quantify nutrient loading during fallow conditions and to establish pretreatment conditions for the study watersheds.

Management within each land use, cultivated and pasture, was consistent to minimize confounding differences due to differing management. Management for the cultivated watersheds, each with contour broad-base terraces and a grassed waterway, consisted of tillage, planting, harvest, and application of litter, supplemental N, and pesticides (Table 3). Management for the pastures consisted of annual litter application, hay harvest (or grazing), and herbicide application (Table 3). The grazed pasture watershed (SW17) was opened for selective grazing during the periods indicated in Table 3 with an

Table 2. Selected characteristics of edge-of-field and downstream study watersheds.

Characteristic	Cultivated watersheds					
	Y6	Y8	Y10	Y13	W12	W13
Area, ha	6.6	8.4	7.5	4.6	4.0	4.6
Slope, %	3.2	2.2	1.9	2.3	2	1.1
Curve number	87	87	87	87	87	87
Litter rate, Mg ha^{-1} yr^{-1}	0.0	13.4	6.7	4.5	9.0	11.2
Mean N rate, kg ha^{-1} yr^{-1}	168	370	278	237	296	328
Mean P rate, kg ha^{-1} yr^{-1}	19	358	196	122	229	286
Land use/crop						
2001	fallow	fallow	fallow	fallow	fallow	fallow
2002	corn	corn	corn	corn	corn	corn
2003	corn	corn	corn	corn	corn	corn
Mehlich-3 P, mg kg^{-1}						
2001	20.1	15.2	19.7	19.1	22.2	19.8
2002	20.9	51.7	40.9	43.5	55.1	68.3
2003	17.7	91.2	63.9	45.0	62.6	111.2
	Pasture watersheds					
	SW12	SW17	Y14	W10		
Area, ha	1.2	1.2	2.3	8.0		
Slope, %	1.6	2.5	3.7	2.5		
Curve number	78	78	76	80		
Litter rate, Mg ha^{-1} yr^{-1}	0.0	0.0	13.4	6.7		
Mean N rate, kg ha^{-1} yr^{-1}	0	0	328	172		
Mean P rate, kg ha^{-1} yr^{-1}	0	0	339	180		
Land use	native prairie	coastal bermudagrass (grazed)	Kleingrass	coastal bermudagrass		
Mehlich-3 P, mg kg^{-1}						
2001	18.0	15.9	10.8	15.9		
2002	15.6	12.6	66.5	37.6		
2003	4.4	5.5	71.7	36.6		
	Downstream watersheds					
	Y	Y2	W1			
Area, ha	125	53	71			
Slope, %	2.4	2.6	2.2			
Mean N rate, kg ha^{-1} yr^{-1}	60	140	91			
Mean P rate, kg ha^{-1} yr^{-1}	37	87	34			
Cultivated, %	33	56	30			
Improved pasture, %	6	4	32			
Rangeland, %	31	39	38			

estimated range of 11.8 to 23.4 kg ha^{-1} of manure deposited daily.

The first annual litter application on cultivated and pasture watersheds occurred in July 2001 and the second in September 2002. At the time of application, litter samples were collected for analysis (Table 4). For the cropped watersheds, target available N rates were set at approximately 170 kg ha^{-1}. This is a typical N rate for corn production in the area and follows crop production recommendations (Gass, 1987). It was assumed that 40% of litter N was available the first year following application and that 10% was available in the second year. Supplemental N (as urea ammonium nitrate [UAN], 50% liquid urea and 50% ammonium nitrate) was applied before planting to reach the 170 kg ha^{-1} N target. No supplemental P was applied in the first litter application year, but 38 kg ha^{-1} of P was added to the control watershed (Y6) in the second litter application year. This practice of alternate annual P application is typical for the Blackland Prairie. For the pasture watersheds, no supplemental N or P was applied.

Annual soil samples were taken in each watershed each winter with a manual soil probe (2.54 cm in diameter). The 15-cm-deep cores, taken at the frequency of at least one core per 0.4 ha, were composited for each watershed. The samples were analyzed for extractable P by the Mehlich-3 procedure (Mehlich, 1984) and the data are presented in Table 2. The soil samples were also analyzed for total Kjeldahl N and P, total and organic C, and micronutrients (data not presented). Crop yield data were collected for the cultivated watersheds and dry weight biomass data for the pasture watersheds.

Water Quality Sampling and Analysis

The outlet of each watershed was equipped with a flow control structure, either a v-notch weir or a flume and weir combination depending on watershed size. Flow data for this study were recorded continuously for these ephemeral channels on 5- to 15-min intervals depending on watershed size. An automated sampler with programmable operation, sample collection pump, and sample bottles was also installed at each flow control structure. Variable time-weighted, discrete samples were collected automatically during runoff events. This sampling strategy collects more samples early in the event to adequately capture the first flush and fewer samples later to adequately sample throughout the event duration. The time durations between samples were similar for similar-sized watersheds and were selected based on previous experience with the site. The sampling strategy for the edge-of-field sites was in general as follows: the first sample was taken after 5 min, then four samples were taken on 15-min intervals, four on 30-min intervals, four on 60-min intervals, and 11 on 120-min intervals. For larger watersheds, the number of samples taken at shorter intervals was reduced, and the durations between samples were increased. Tipping bucket and standard rain gauges measured rainfall depth and intensity data.

Water quality samples were collected from the field within

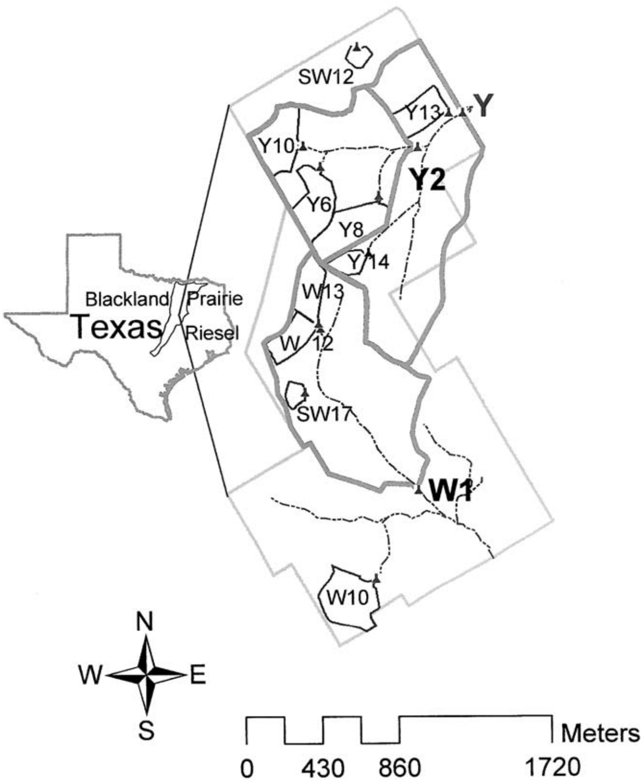


Fig. 1. The location of edge-of-field and downstream watersheds used in this study.

48 h of each runoff event. Collected samples were acidified with concentrated HCl, iced, and transported to the laboratory for analysis of dissolved and sediment-bound nutrients. Samples were stored at 4°C before analysis. Samples were analyzed for dissolved nitrate plus nitrite N ($NO_3 + NO_2$–N), ammonia N (NH_4–N), and ortho-phosphate (PO_4–P) concentrations using a Technicon Autoanalyzer IIC (Bran-Luebbe, Roselle, IL) and colorimetric methods published by Technicon Industrial Systems (1973a, 1973b, 1976). Results for $NO_3 + NO_2$–N are reported as NO_3–N because the NO_3–N form dominates. Dissolved N and P loads were then determined by multiplying the concentration in each discrete sample by the corresponding flow volume and summing these incremental loads for the runoff event duration. The sediment concentration in each discrete sample was determined by allowing the sample settle for 3 to 5 d and decanting off a majority of the solution. The sediment slurry was dried at 116°C for 18 to 24 h, and the

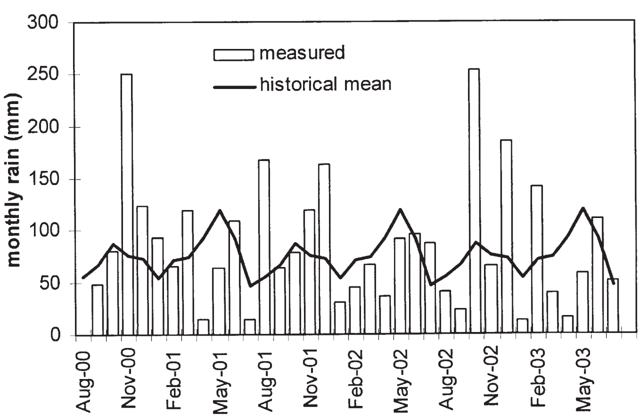


Fig. 2. Measured monthly precipitation for the three-year study period.

mass of sediment was then determined. This mass divided by the measured volume of collected sample represented the sediment concentration. As with dissolved nutrients, sediment load for each storm event was estimated as the sum of incremental runoff volumes multiplied by the corresponding sediment concentration. Sediment-bound total Kjeldahl N and P levels were determined by a salicylic acid modification of a semimicro-Kjeldahl digestion procedure (Technicon Industrial Systems, 1976). Sediment-bound N and P loads were then determined by multiplying the nutrient concentrations by corresponding sediment masses.

Experimental Design and Statistical Analysis

In this study, the 10 edge-of-field watersheds were the experimental units. The treatments included land use (cultivated and pasture) and litter rate from 0.0 to 13.4 Mg ha^{-1}. Special effort was made to keep management activities the same within each land use category, so results were not confounded by differing management. To evaluate the impact of conversion to a poultry litter fertilization strategy, the presence of inherent linear relationships between the proposed litter rates assigned to the study watersheds and water quality before litter application were examined using linear regression. Specifically, possible linear relationships were examined between litter rate and annual mean, median, and maximum NO_3–N, NH_4–N, and PO_4–P concentrations and annual N and P loads. The presence of inherent relationships before treatment would increase the complexity in treatment effect analysis. For all regression analyses, the linear relationship was judged significant if the slope of the regression line was significantly different than zero based on an a priori $\alpha = 0.05$ probability level. Possible differences in annual nutrient loads, specifically median loads, were also analyzed with Mann–Whitney tests. This nonparametric test was used because intra-annual storm load data were not normally distributed as determined by the Kolmogorov–Smirnov test ($\alpha = 0.05$). All statistical tests were conducted with Minitab software (Minitab, 2000) and according to procedures described in Helsel and Hirsch (1993) or Haan (2002).

RESULTS AND DISCUSSION

Nutrient Concentrations in Runoff from Edge-of-Field Watersheds

In the first study year, in which water quality was monitored under fallow conditions with no applied fertilizer, 11 runoff events occurred and were monitored. Nutrient loads and concentrations were relatively low as was expected with no applied fertilizer. The mean annual concentrations of discrete samples collected at edge-of-field sites were all less than 1.2 mg L^{-1} for NO_3–N, 0.1 mg L^{-1} for NH_4–N, and 0.4 mg L^{-1} for PO_4–P (Table 5). Annual N and P loads were all less than 6 and 2 kg ha^{-1}, respectively (Tables 6 and 7).

During this fallow year before litter application, no significant relationships between water quality and proposed litter rates assigned to the study watersheds with similar land use were determined. Specifically, no significant linear relationships were determined between proposed litter rate and mean, median, and maximum annual NO_3–N, NH_4–N, and PO_4–P concentrations determined from the entire set of discrete storm samples for the pretreatment year (Table 8). In addition, no significant

Table 3. Management activities for the cultivated and pasture watersheds.

Year	Date	Study year	Management activity
		Cultivated watersheds	
2000	1. Aug.	fallow	study period began
	3–8 Aug.	(Year 1)	corn harvest, shred stalks (Y6, Y13, W12, W13)
	14 Aug.–22 Sept.		tillage
	2–4 Oct.		tillage
	11–13 Oct.		terrace work (Y10, Y13, W12, W13)
2001	27 Mar.–27 Apr.		tillage
	29 May–1 June		tillage
	11–17 July		poultry litter application and incorporation
	18–21 Sept.	first-year litter	herbicide application
	26–28 Sept.	(Year 2)	tillage
	29–30 Oct.		tillage
	2 Nov.		herbicide application
2002	20–21 Feb.		supplemental fertilizer application and incorporation
	6–7 Mar.		corn planting
	11 Mar.		herbicide application
	22–24 Apr.		tillage
	19–24 Aug.	second-year litter	corn harvest
	28–30 Aug.	(Year 3)	shred stalks
	3–5 Sept.		poultry litter application and incorporation
	23–27 Sept.		tillage
2003	30–31 Jan.		supplemental fertilizer application and incorporation
	17–19 Mar.		tillage
	17–20 Mar.		corn planting
	17–20 Mar.		herbicide application
	29 Apr.		pesticide application
	20–25 Aug.		corn harvest
		Pasture watersheds	
2000	1 Aug.	fallow	study period began
	Oct.–Dec.	(Year 1)	grazed (SW17)
2001	Mar.–July		grazed (SW17)
	12–19 Apr.		herbicide application
	22 May–10 July		hay harvest (Y14, SW12, W10)
	13 July		poultry litter application (W10, Y14)
	3–4 Oct.	first-year litter	hay harvest (W10, Y14)
	Oct.–Dec.	(Year 2)	grazed (SW17)
2002	15 Mar.–5 July		grazed (SW17)
	5–28 Apr.		herbicide application
	28 June–12 July		hay harvest (SW12, Y14, W10)
	5 Sept.	second-year litter	poultry litter application (W10, Y14)
	15 Oct.–31 Jan. 2003	(Year 3)	grazed (SW17)
2003	1 Mar.–30 July		grazed (SW17)
	23–26 Apr.		herbicide application (SW17, W10, Y14)
	29 May–24 June		hay harvest (SW12, W10, Y14)

relationships were determined between proposed litter rate and annual N and P loads. Therefore, any relationships between nutrient concentrations and loads and litter rate that occurred after the first litter application were confidently attributed to treatment effect.

Nutrient concentrations in the first year following litter application were generally greater than those measured under fallow conditions with no fertilizer applied (Table 5). In this first application year that produced 16 runoff events, increased NO_3–N and NH_4–N concentrations were measured from edge-of-field watersheds with applied litter and with inorganic fertilizer only; therefore, the increases are not attributed specifically to litter application. For example, the mean annual NO_3–N concentration increased by 11.1 mg L^{-1} for Y6 with UAN application and by 3.9 to 13.2 mg L^{-1} for the litter application watersheds. Based on linear regression of litter rate and annual mean, median, and maximum NO_3–N and NH_4–N concentrations, litter rate did not affect concentrations of dissolved N constituents in runoff from the cultivated or pasture watersheds (Table 8). These results, which are illustrated by event mean concentrations (EMCs) in Fig. 3a, are not surprising for the cultivated watersheds as available N application rates were similar.

In contrast to N results, increased PO_4–P concentrations (+0.07 to 0.47 mg L^{-1}) were measured from edge-of-field watersheds with applied litter but not from the control watershed in which no organic or inorganic P was applied. Annual mean, median, and maximum PO_4–P concentrations all exhibited significant relationships with litter rate for both cultivated and pasture water-

Table 4. Results from poultry litter analysis. Data presented are means of four replications with standard deviations in parentheses.

Litter application date	Total N	Total P	Water-extractable nutrients			Organic C
			NO_3–N	NH_4–N	SRP†	
	%	%	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	%
July 2001	2.32 (0.33)	2.14 (0.12)	211 (245)	1170 (370)	895 (238)	28.4 (6.3)
September 2002	3.05 (0.24)	3.47 (0.47)	857 (293)	3775 (8)	1233 (35)	31.2 (0.6)

† Soluble reactive phosphorus.

Table 5. Annual maximum, mean, and median nutrient concentrations in surface runoff.

Watershed	n†	Litter rate	NO_3–N concentration			NH_4–N concentration			PO_4–P concentration		
			Maximum	Mean	Median	Maximum	Mean	Median	Maximum	Mean	Median
		Mg ha^{-1}	mg L^{-1}								
Cultivated watersheds, fallow year											
Y6	14	0.0	1.04	0.60	0.58	0.07	0.02	0.02	0.40	0.31	0.32
Y13	103	0.0	3.35	1.14	1.12	0.14	0.03	0.02	0.68	0.33	0.32
Y10	29	0.0	2.63	0.87	0.52	0.13	0.02	0.02	0.85	0.34	0.29
W12	75	0.0	1.99	0.84	0.82	0.19	0.04	0.02	0.45	0.20	0.18
W13	58	0.0	1.82	0.69	0.62	0.14	0.03	0.02	0.50	0.23	0.21
Y8	48	0.0	1.53	0.82	1.05	0.17	0.03	0.03	0.50	0.29	0.26
Cultivated watersheds, first-year litter											
Y6	76	0.0	78.03	11.70	2.75	2.53	0.30	0.02	0.54	0.23	0.24
Y13	101	4.5	78.88	10.55	2.20	1.52	0.19	0.05	0.94	0.40	0.35
Y10	113	6.7	90.10	14.08	6.04	1.70	0.20	0.06	1.13	0.58	0.58
W12	87	9.0	57.12	7.07	0.91	1.39	0.24	0.05	2.15	0.52	0.43
W13	81	11.2	30.30	4.61	0.48	1.12	0.15	0.05	3.25	0.69	0.56
Y8	96	13.4	56.27	11.01	1.92	1.94	0.29	0.11	3.22	0.76	0.65
Cultivated watersheds, second-year litter											
Y6	78	0.0	88.60	25.66	3.38	6.38	0.96	0.05	0.51	0.23	0.22
Y13	88	4.5	87.50	18.67	3.56	1.86	0.22	0.06	0.87	0.44	0.41
Y10	102	6.7	84.00	23.49	7.57	2.77	0.30	0.05	1.23	0.77	0.81
W12	59	9.0	51.50	14.13	7.67	0.36	0.10	0.06	1.02	0.63	0.59
W13	64	11.2	42.10	13.21	9.00	0.36	0.08	0.07	1.89	1.11	1.09
Y8	85	13.4	45.70	15.60	9.43	0.40	0.09	0.06	1.78	1.01	0.93
Pasture watersheds, fallow year											
SW12	17	0.0	0.28	0.05	0.02	0.06	0.02	0.02	0.32	0.14	0.11
SW17	15	0.0	0.77	0.12	0.04	0.08	0.04	0.04	0.27	0.20	0.19
W10	13	0.0	0.23	0.12	0.12	0.37	0.07	0.04	0.24	0.15	0.14
Y14	14	0.0	0.28	0.08	0.08	0.09	0.03	0.02	0.55	0.15	0.12
Pasture watersheds, first-year litter											
SW12	53	0.0	0.18	0.04	0.01	0.08	0.04	0.03	0.25	0.08	0.07
SW17	34	0.0	0.42	0.10	0.06	0.19	0.07	0.06	0.27	0.19	0.18
W10	32	6.7	1.26	0.24	0.05	0.12	0.07	0.06	0.87	0.66	0.62
Y14	14	13.4	0.01	0.00	0.00	0.16	0.06	0.06	1.77	1.29	1.27
Pasture watersheds, second-year litter											
SW12	97	0.0	0.34	0.04	0.03	1.10	0.05	0.02	0.88	0.09	0.07
SW17	102	0.0	1.38	0.11	0.02	0.11	0.02	0.02	0.45	0.18	0.17
W10	59	6.7	0.47	0.11	0.07	0.08	0.05	0.04	1.56	1.09	1.06
Y14	19	13.4	0.32	0.13	0.13	0.12	0.06	0.05	3.96	2.29	2.44
Downstream watersheds, fallow year											
Y	82	0.0	9.10	1.76	0.87	0.13	0.04	0.03	0.43	0.20	0.19
Y2	NA‡	0.0	NA	NA	NA	NA	NA	NA	NA	NA	NA
W1	64	0.0	2.94	0.60	0.22	0.47	0.05	0.03	0.83	0.26	0.21
Downstream watersheds, first-year litter											
Y	52	1.7	12.66	2.92	0.76	0.47	0.12	0.09	0.75	0.27	0.21
Y2	103	3.1	34.85	6.44	2.11	0.72	0.12	0.05	0.78	0.41	0.38
W1	76	0.3	8.52	1.85	0.58	0.29	0.05	0.03	0.45	0.23	0.20
Downstream watersheds, second-year litter											
Y	128	1.7	17.00	4.78	3.59	0.45	0.08	0.05	0.48	0.30	0.30
Y2	116	3.1	33.20	6.38	3.11	1.23	0.10	0.06	0.80	0.39	0.39
W1	85	0.3	6.28	1.04	0.36	0.28	0.04	0.03	0.47	0.22	0.21

† Annual number of discrete samples taken during runoff events.
‡ Samples were not analyzed for dissolved nutrients.

sheds (Table 8). These relationships are illustrated with EMCs in Fig. 4a. Although the present study and others, for example Kleinman and Sharpley (2003), have reported significant positive correlation between manure application rate and P concentrations in runoff, the relative contribution of P from applied manure and from P in the upper soil layer is not well established.

Nutrient concentrations were generally greater in the second year following litter application compared with the previous year (Table 5). In this second litter application year with 13 runoff events, litter rate affected concentrations of both N and P constituents. On the cultivated watersheds, annual mean and maximum NO_3–N and NH_4–N concentrations decreased as litter rate increased (Table 8). These significant relationships are illustrated by EMCs for NO_3–N in Fig. 3b. Similar results were obtained by Vories et al. (2001) who reported higher NO_3–N loads and concentrations for cotton fields fertilized with UAN compared with poultry litter, and by Edwards and Daniel (1994) who reported higher NO_3–N and NH_4–N concentrations from pasture plots with inorganic fertilizer compared with poultry litter. Results in this study can be explained by the higher proportion of available NO_3–N and NH_4–N applied as supplemental inorganic N on watersheds with lower litter rates and by greater amounts of low mobility organic N on watersheds with high litter rates. Whereas organic fertilizers are dominated by organic N forms that must be mineral-

Table 6. Annual total, dissolved, and sediment-bound N loads for the edge-of-field and downstream watersheds.†

Site	Litter rate	Total N load Fallow	Total N load First-year litter	Total N load Second-year litter	Dissolved N Fallow	Dissolved N First-year litter	Dissolved N Second-year litter	Sediment-bound N Fallow	Sediment-bound N First-year litter	Sediment-bound N Second-year litter
	Mg ha^{-1}	kg ha^{-1}								
Cultivated watersheds										
Y6	0.0	1.2	22.5	42.4	0.3	14.7	38.8	0.9	7.9	3.6
Y13	4.5	5.7	50.5	32.6	2.1	28.5	29.3	3.5	22.0	3.3
Y10	6.7	2.2	41.3	52.8	0.7	34.7	50.3	1.6	6.6	2.5
W12	9.5	4.3	28.0	18.7	1.1	13.0	16.0	3.2	14.9	2.7
W13	11.2	2.6	18.3	27.5	0.9	7.2	24.7	1.8	11.1	2.8
Y8	13.4	1.9	28.6	24.1	0.6	16.1	21.7	1.2	12.5	2.4
Pasture watersheds										
SW12	0.0	NA‡	0.6	0.7	NA	0.1	0.2	NA	0.5	0.5
SW17	0.0	NA	8.5	0.4	NA	0.1	0.2	NA	8.4	0.2
W10	6.7	0.1	0.3	0.2	0.1	0.2	0.1	0.0	0.1	0.1
Y14	13.4	0.0	0.1	0.1	0.0	0.0	0.1	0.0	0.1	0.0
Downstream watersheds										
Y	1.7	4.0	6.3	8.7	2.7	4.0	7.4	1.2	2.4	1.2
Y2	3.1	NA	22.5	18.8	NA	13.0	17.0	NA	9.5	1.8
W1	0.3	2.2	4.2	3.6	0.8	2.0	2.2	1.3	2.1	1.4

† Differences are due to rounding.
‡ A complete set of storm samples was not collected; therefore, annual loads were not determined.

Table 7. Annual total, dissolved, and sediment-bound P loads for the edge-of-field and downstream watersheds.†

Site	Litter rate	Total P load Fallow	Total P load First-year litter	Total P load Second-year litter	Dissolved P Fallow	Dissolved P First-year litter	Dissolved P Second-year litter	Sediment-bound P Fallow	Sediment-bound P First-year litter	Sediment-bound P Second-year litter
	Mg ha^{-1}	kg ha^{-1}								
Cultivated watersheds										
Y6	0.0	0.4	3.2	1.7	0.3	0.4	0.5	0.2	2.8	1.3
Y13	4.5	1.7	10.6	1.9	1.1	1.3	0.9	0.6	9.3	1.0
Y10	6.7	0.6	3.4	2.8	0.4	1.4	2.1	0.3	1.9	0.7
W12	9.5	1.5	7.9	1.6	1.2	1.2	0.8	0.3	6.7	0.9
W13	11.2	0.7	5.9	3.1	0.6	1.2	2.1	0.2	4.7	1.0
Y8	13.4	0.5	6.6	2.4	0.3	1.4	1.6	0.2	5.1	0.8
Pasture watersheds										
SW12	0.0	NA‡	0.1	0.2	NA	0.1	0.1	NA	0.0	0.1
SW17	0.0	NA	1.9	0.3	NA	0.2	0.3	NA	1.7	0.0
W10	6.7	0.0	0.6	1.2	0.0	0.5	1.1	0.0	0.0	0.0
Y14	13.4	0.0	0.4	0.6	0.0	0.4	0.6	0.0	0.0	0.0
Downstream watersheds										
Y	1.7	0.6	1.2	0.8	0.2	0.3	0.6	0.4	0.9	0.3
Y2	3.1	NA	3.7	1.3	NA	0.8	0.9	NA	2.9	0.4
W1	0.3	0.7	0.8	0.7	0.2	0.3	0.4	0.4	0.5	0.3

† Differences are due to rounding.
‡ A complete set of storm samples was not collected; therefore, annual loads were not determined.

ized before becoming plant available, inorganic fertilizers contain N forms that are readily solubilized. Mean, median, and maximum PO_4–P concentrations again increased as litter rate increased for both cultivated and pasture watersheds (Table 8). These relationships are illustrated by EMCs of PO_4–P in Fig. 4b.

Concentration Changes over Time

In each of the cultivated watersheds that received poultry litter, annual mean NO_3–N and PO_4–P concentrations in runoff increased each year from the fallow year to the second year of litter application (Table 5). In contrast, for watershed Y6 with no litter but annual inorganic N fertilizer application and inorganic P application in 2003, PO_4–P concentration in runoff decreased over the period. Results for the cultivated control (Y6) and the cultivated watershed with 9.0 Mg ha^{-1} annual litter rate (W12) were chosen to illustrate the difference in runoff nutrient concentrations between the watersheds with inorganic fertilizer only and the watersheds with litter application (Fig. 5 and 6).

Soil Phosphorus and Phosphorus Concentrations in Runoff

Following litter application, PO_4–P concentrations in runoff were positively correlated to Mehlich-3 soil P levels with r^2 values ranging from 0.43 to 1.00. Several studies have also related soil P levels to P losses and concentrations in runoff (e.g., Sharpley and Smith, 1992; Sharpley et al., 1999; Kleinman et al., 2000; Torbert et al., 1996). In one study of particular relevance because it was also conducted on Houston Black clay soil, Torbert et al. (2002) reported significant correlation between dissolved P in runoff and both water-extractable and Mehlich-3 soil P. In the present study, watershed soils contained low Mehlich-3 P levels (10.8–22.2 mg kg^{-1})

Table 8. Regression results between litter rate and annual mean, median, and maximum NO_3–N, NH_4–N, and PO_4–P concentrations in runoff at the edge-of-field watersheds.

	NO_3–N		NH_4–N		PO_4–P	
	p	r^2	*p*	r^2	*p*	r^2
			Cultivated watersheds, fallow year			
Mean	0.935	0.00	0.276	0.28	0.328	0.24
Median	0.585	0.08	0.203	0.37	0.138	0.46
Maximum	0.937	0.00	0.061	0.63	0.954	0.00
			Cultivated watersheds, first-year litter			
Mean	0.390	0.19	0.684	0.05	0.001***	0.94
Median	0.490	0.13	0.052	0.65	0.018*	0.79
Maximum	0.144	0.45	0.232	0.33	0.005**	0.88
			Cultivated watersheds, second-year litter			
Mean	0.048*	0.67	0.027*	0.74	0.008**	0.86
Median	0.007**	0.87	0.157	0.43	0.019*	0.78
Maximum	0.019*	0.79	0.016*	0.80	0.010**	0.84
			Pasture watersheds, fallow year			
Mean	0.974	0.00	0.839	0.03	0.614	0.15
Median	0.333	0.45	0.698	0.09	0.609	0.15
Maximum	0.480	0.27	0.763	0.06	0.245	0.57
			Pasture watersheds, first-year litter			
Mean	0.851	0.02	0.754	0.06	0.004**	0.99
Median	0.527	0.22	0.478	0.27	0.006**	0.99
Maximum	0.923	0.01	0.800	0.04	0.005**	0.99
			Pasture watersheds, second-year litter			
Mean	0.316	0.47	0.296	0.50	0.002**	0.93
Median	0.006**	0.99	0.014*	0.97	0.006**	0.99
Maximum	0.469	0.28	0.485	0.27	0.035*	0.93

* Significant at the 0.05 probability level.
** Significant at the 0.01 probability level.
*** Significant at the 0.001 probability level.

in the top 15 cm before litter application, but litter application raised Mehlich-3 soil P to 36.6 to 111.2 mg kg^{-1} levels (Table 2). The amount of P applied to induce a 1.1 kg ha^{-1} change in soil P ranged from 2 to 46 kg ha^{-1} with a median of 3.9 kg ha^{-1}. Thus, P application did create substantial increases in soil P on these sites with low initial soil P levels. These increases are comparable with those experienced on the same soil by Torbert et al. (2002), who used a ratio of 3:1 ratio in building soil test P to desired levels. The largest increases in soil P occurred in watersheds with the greatest application rate (Fig. 7), which is similar to trends in runoff PO_4–P concentrations shown in Fig. 4. It is expected that the increases in PO_4–P runoff concentrations, which coincided with increases in Mehlich-3 soil P levels, will continue to occur if P in excess of crop requirements is annually added to these watersheds.

Influence of Management Practices

A commonly accepted best management practice is to avoid fertilizer application immediately before rainfall. Recent research has illustrated that the timing of rainfall events following litter and inorganic fertilizer application is an important determinant of nutrient loss, specifically higher nutrient concentrations in runoff events soon after application (Kleinman and Sharpley, 2003; Pierson et al., 2001; Eghball et al., 2002). Results in this study demonstrate that inorganic fertilizer, in this case UAN, can be much more susceptible than poultry litter to NO_3–N (Fig. 5) and NH_4–N loss (not shown) in rains immediately following fertilizer application. Even so,

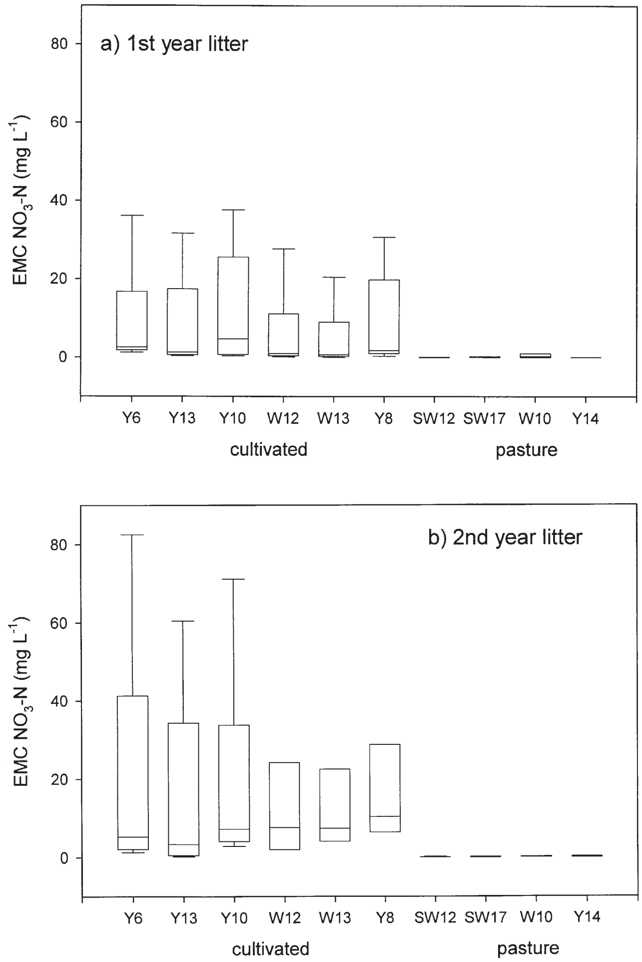


Fig. 3. Event mean NO_3–N concentrations for the cultivated and pasture watersheds in the (a) first litter application year and (b) second litter application year. Litter rate increases from left to right, within each land use group, on the *x* axis.

care should be taken to avoid litter application before heavy rainfall forecasts to reduce the risk of offsite nutrient transport. High dissolved P concentrations in runoff are to some extent limited in runoff events soon after application to cultivated fields because P is more readily sorbed to soil than N. Concentrations of PO_4–P in runoff from the pasture watersheds with surface litter application tended to exceed those in runoff from cultivated watersheds, which is due in part to less contact between soil and litter. The split application of N in the watersheds with applied litter and supplemental N also tended to reduce peak NO_3–N concentrations, which is another recommended practice to reduce offsite transport.

Nutrient Loads from Edge-of-Field Watersheds

In contrast to nutrient concentrations, which exhibited several significant relationships with litter rate, no significant linear relationships between litter rate and annual total (sum of dissolved and sediment-bound) N and P loads for the pasture or cultivated watersheds were detected by regression analysis. Variability in runoff and erosion between watersheds tended to mask differences in N and P loads. The application of similar available

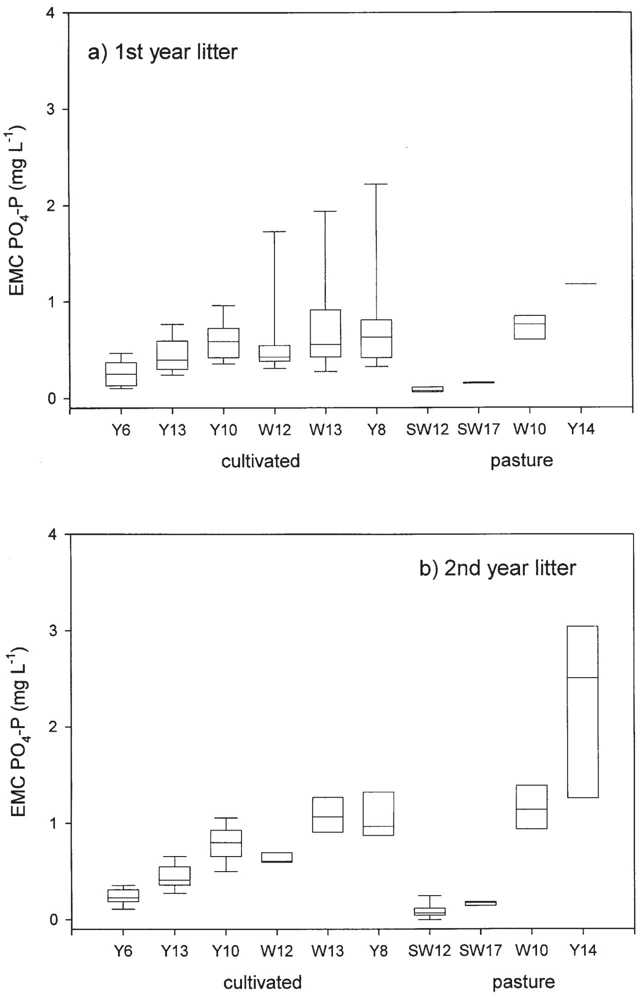


Fig. 4. Event mean PO_4–P concentrations for the cultivated and pasture watersheds in the (a) first litter application year and (b) second litter application year. Litter rate increases from left to right, within each land use group, on the *x* axis.

N rates on cultivated watersheds also contributed to the lack of difference in N loads.

While no consistent relationships between litter rate and annual nutrient loads were evident, potential differences in storm loads within individual years were explored. Since the intra-annual storm load data were not normally distributed as determined by the Kolmogorov–Smirnov test, Mann–Whitney tests were used to determine differences in median storm loads. No significant differences in median storm total N or total P loads were determined in the fallow year or the first year of litter application for the cultivated or pasture watersheds; however, several significant differences were determined in the second year following litter application. For the cultivated watersheds, the median P storm load from the control watershed Y6 (0.08 kg ha^{-1}) was less than from Y10 (0.25 kg ha^{-1}), W13 (0.26 kg ha^{-1}), and Y8 (0.30 kg ha^{-1}) litter, and the median P load from Y13 (0.08 kg ha^{-1}) was less than from W13 (0.26 kg ha^{-1}). Also, the median N storm load was less from Y13 (0.80 kg ha^{-1}) than from Y10 (2.71 kg ha^{-1}). For the pasture watersheds, no significant differences in median N loads were determined in the year following the second litter application, but several differences were determined for P loads. The median P storm load for both pastures with applied litter (W10 = 0.20 kg ha^{-1}, Y14 = 0.08 kg ha^{-1}) exceeded the storm P load for native prairie pasture with no litter SW12 (0.03 kg ha^{-1}), and the median P load from W10 (0.20 kg ha^{-1}) exceeded that of the grazed pasture with no litter SW17 (0.03 kg ha^{-1}).

Load Changes over Time

As expected, nutrient loads were greater for the first year following litter application than the fallow year. The increasing trend, however, did not continue for the cultivated watersheds in the second year of litter application. Annual total N loads were generally similar although both increases and decreases were observed for individual watersheds in the two litter application years (Table 6). In contrast, annual total P loads decreased in every cultivated watershed from the first to the second application year (Table 7). For the cultivated watersheds, sediment-bound P loads decreased dramatically from the first to second year following liter application as mean erosion rates decreased from 7474 to 1123 kg ha^{-1} even though annual runoff volumes were similar as shown in Table 9. Relatively little change in dissolved P loads was observed in these two years. The majority of the annual erosion in the first application year occurred in two major events in which more than 100 mm of rainfall fell within 24 h. On the pasture watersheds that received litter, total N loads were similar in both litter application years, but total P loads increased each year. The exception occurred in the grazed pasture watershed, which experienced unusually high total N and P loads in 2000–2001. These results illustrate the importance of land use impacts on erosion and the need to compare runoff volume and intensity when examining nutrient flux.

Comparison between Cultivated and Pasture Watersheds

The influence of land use on water quality was evaluated by comparing runoff N and P concentrations and storm N and P loads between pasture and cultivated watersheds. In the fallow year, NH_4–N and PO_4–P concentrations in runoff were fairly comparable between land uses, but NO_3–N concentrations were considerably higher in the cultivated than the pasture watersheds, possibly due increased nutrient uptake by the permanent pasture vegetation (Table 5). In the years following litter application, PO_4–P concentrations tended to be higher for the pasture watersheds, which may be due to the litter remaining on the pasture's thatch surface limiting soil and litter interaction and P adsorption. The NO_3–N and NH_4–N concentrations, however, were higher for the cultivated watersheds than for the pasture watersheds, probably due to higher N application rates with supplemental UAN and possibly to increased plant uptake in the pasture watersheds.

In each of the three study years, median N and P storm loads from the pasture watersheds were signifi-

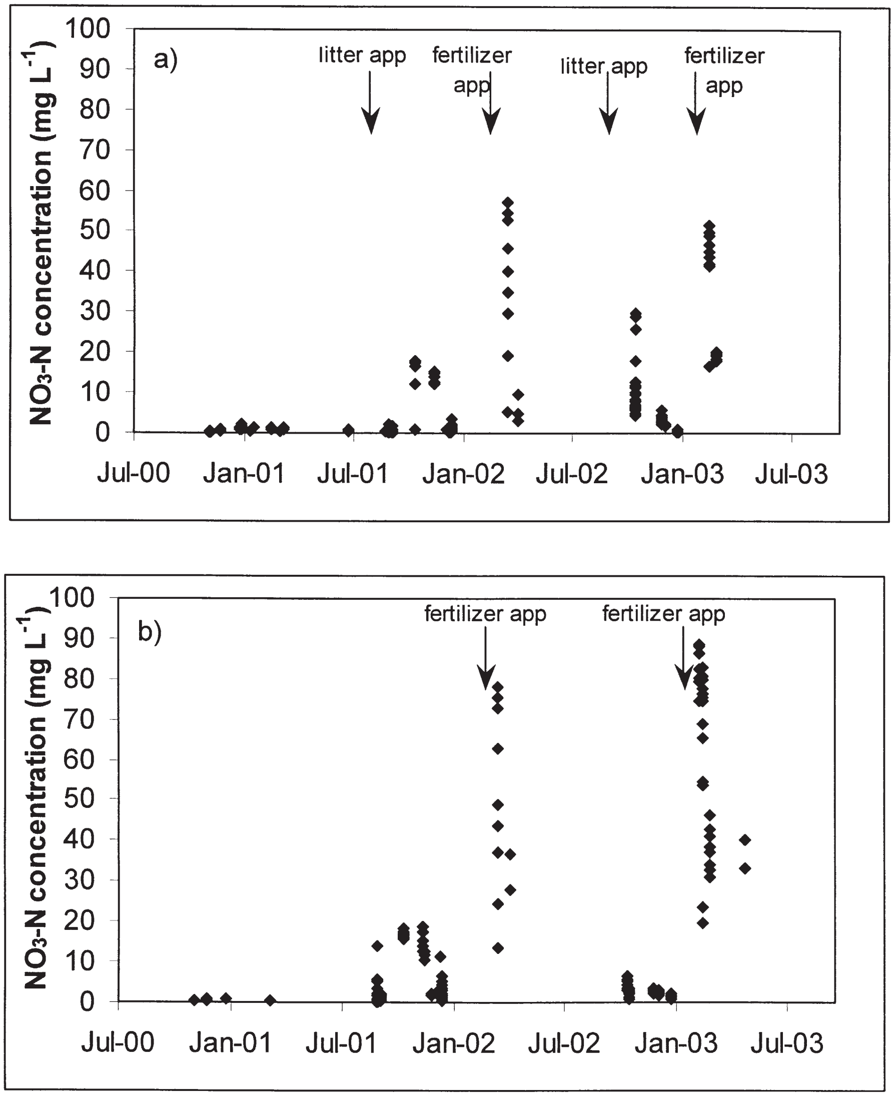
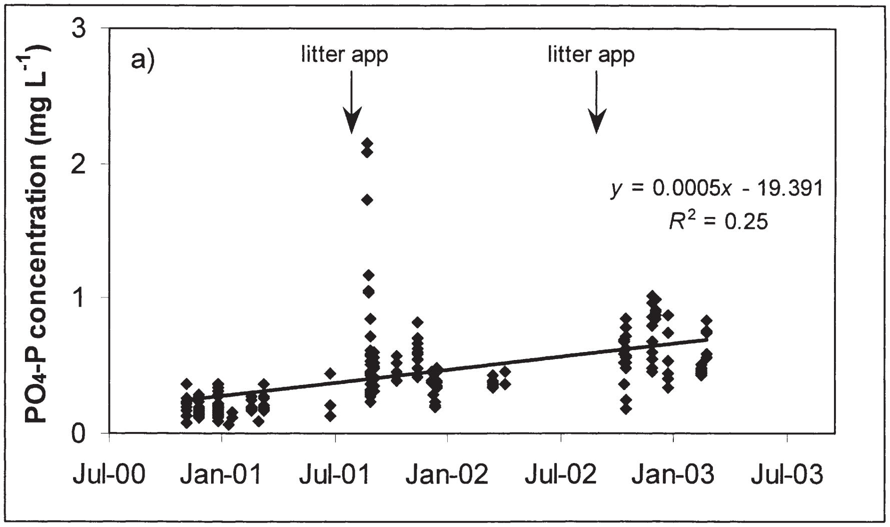


Fig. 5. Comparison of NO_3–N concentration in runoff in relation to time of application for watersheds with different fertilization strategies. Dates of N applications are indicated by arrows. Watershed W12 was chosen to represent the five watersheds with similar fertilization for comparison with the control watershed, Y6. (a) W12 with split litter and supplemental N application. (b) Y6 with a single annual inorganic N application.

cantly lower than from the cultivated watersheds as evaluated with the Mann–Whitney test. This difference is attributed to less runoff and erosion from the pasture watersheds. In the years with inorganic fertilizer and litter application, a majority of N lost from the cultivated watersheds was in the dissolved form (Table 6). The percentage of N loss associated with eroded material, determined as the mean of % annual sediment-bound N loss values from the six cultivated watersheds, decreased from the first application year (42%) to the second application year (10%). In contrast to N loss, a larger percentage of P loss was associated with eroded material (79%) in the first litter application year but less so (46%) in the second application year (Table 7). This decrease is attributed to lower erosion rates in the second application year (Table 9). For pasture watersheds, 79 and 43% of the N load was associated with eroded material in the two litter application years; however, 78 and 87% of the P load was in the dissolved form in the same years. In general, the pasture watersheds experienced much lower erosion rates than the cultivated watersheds. An exception occurred in the grazed pasture in 2001–2002. Also, much of the eroded material in the pasture consisted of plant residue and litter.

The measured loads in this study are especially interesting in comparison with results from similar previous field-scale studies described in Table 1. Annual N and P losses from the pasture watersheds were less than those measured on pastures by Edwards et al. (1996) and Pierson et al. (2001), but P loads were similar to those reported by Vervoort et al. (1998). Edwards et al. (1996) reported higher losses, which were probably due to higher soil P levels and deposition of manure by grazing dairy cattle. Pierson et al. (2001) also reported higher P concentrations and loads in two years following litter application, even though runoff volumes and background soil P levels were similar. In contrast, nutrient losses measured from cultivated watersheds in this study



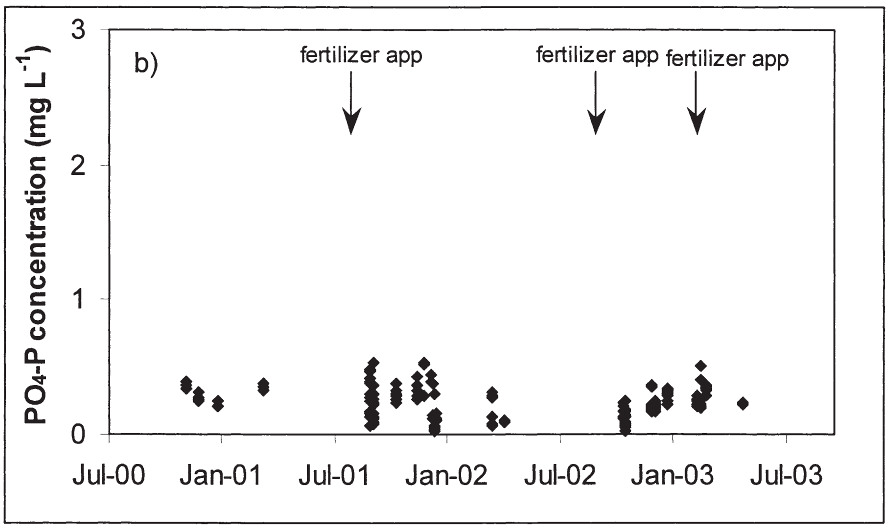


Fig. 6. Comparison of PO_4–P concentration in runoff in relation to time of application for watersheds with different fertilization strategies. Dates of P applications are indicated by arrows. Watershed W12 was chosen to represent the five watersheds with similar fertilization for comparison with the control watershed, Y6. (a) W12 with annual litter application. (b) Y6 with an alternate annual P application.

were considerably higher than in comparable studies. Chinkuyu et al. (2002) reported lower mean annual NO_3–N and PO_4–P loads in surface runoff probably due to subsurface drainage, which contributed to reduced runoff volumes. Hall (1994) and Wood et al. (1999) also reported lower N and P loads, which can be partially attributed to a winter cover crop. Vories et al. (2001) measured lower N and P loads, which can be explained by much smaller runoff amounts.

Influence of Fertilizer Type on Cultivated Watersheds

Two previous studies on the same cultivated watersheds at Riesel reported nutrient loss data under inorganic fertilizer application (Chichester and Richardson, 1992; Sharpley et al., 1987). These previous studies reported lower N loads than reported in the present study, which resulted from lower total N application rates (Fig. 8); however, similar P loads were reported in those

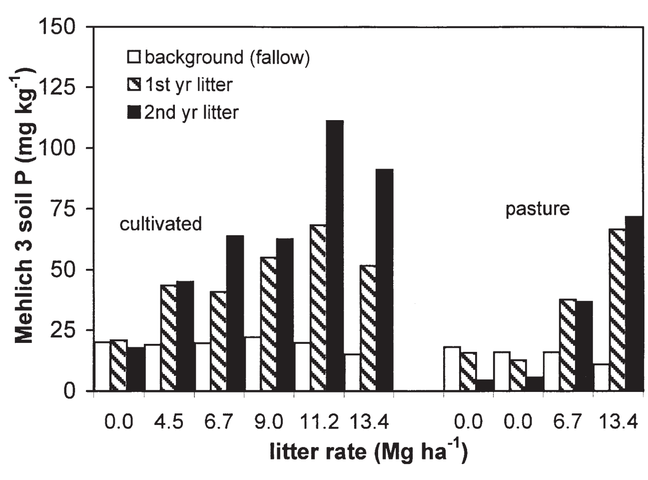


Fig. 7. Changes in soil P levels as a function of litter application rate.

Table 9. Annual runoff depth and sediment loss for edge-of-field and downstream watersheds.

	Runoff depth			Sediment loss		
Watershed	Fallow	First-year litter	Second-year litter	Fallow	First-year litter	Second-year litter
	mm			kg ha^{-1}		
			Cultivated watersheds			
Y6	176	253	273	337	3 661	1 336
Y13	254	353	257	1 898	12 191	1 252
Y10	309	325	350	767	2 930	1 109
W12	200	354	181	2 730	10 911	1 138
W13	132	275	216	1 267	8 490	1 041
Y8	216	243	216	626	6 661	863
			Pasture watersheds			
SW12	212	175	225	0	35	85
SW17	145	129	193	0	1 608	25
W10	166	126	131	5	38	19
Y14	58	65	65	10	50	5
			Downstream watersheds			
Y	295	212	250	284	966	356
Y2	262	223	261	785	1 709	486
W1	199	165	191	470	855	406

studies and from the inorganic-only control watershed (Y6) in this study (Fig. 9).

In the present study, N and P losses were evaluated under three different fertilizer scenarios: (i) fallow with no fertilizer, (ii) poultry litter supplemented with inorganic fertilizer to balance available N across treatments, and (iii) inorganic-only control. In this study, cultivated watersheds that received litter and supplemental N experienced comparable N loads to the control watershed that received only inorganic N (Table 6). These same watersheds, however, had higher P loads than the control watershed, which received less P fertilizer (Table 7). As P application continues in excess of crop needs, which is most pronounced on the high litter rate watersheds, P loads should continue to exceed those from the control watershed.

The relationship between N and P losses and application rates is interesting in terms of fertilization with organic fertilizer, poultry litter in this case (Fig. 8 and 9). Inorganic fertilizer is typically applied with the assumption that N and P is entirely plant available, but organic fertilizers are applied with the assumption that only a portion of the N and P are available depending on mineralization (Mengel and Kirkby, 1978). Therefore, application rates for organic fertilizers are based on available N or P, which results in greater total application compared with inorganic fertilizers.

Downstream Results

As stated by Gburek et al. (2000), edge-of-field nutrient losses should be evaluated in relation to a stream or downstream waterbody because the effects of nutrient losses are experienced downstream. Analyses of results for watersheds with multiple land uses, however, are not as straightforward because of spatial differences in land use patterns. In spite of these differences, several interesting results were observed. As shown in Table 5, the three downstream watersheds experienced lower nutrient concentrations than the edge-of-field watersheds. For example, following litter application mean annual PO_4–P concentrations downstream ranged from 0.22 to 0.41 mg L^{-1} compared with 0.40 to 1.11 mg L^{-1} at the edge-of-field. The downstream watersheds also had lower nutrient loads, on a per area basis (Tables 6 and 7). Lower nutrient losses at the downstream sites are influenced by overall lower nutrient application rates than those applied to the smaller edge-of-field watersheds (Table 2). Nutrient adsorption, transforma-

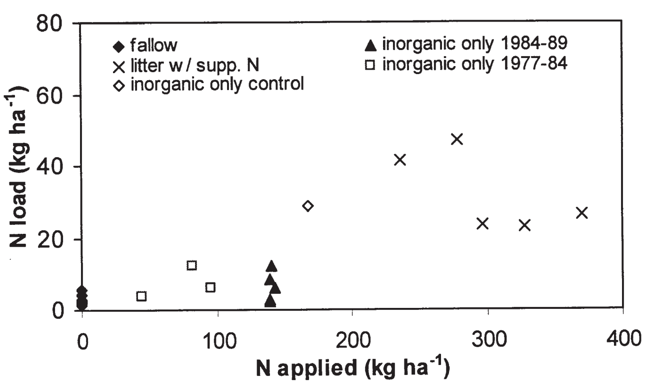


Fig. 8. Mean annual total N loads for the edge-of-field watersheds at Riesel. Loads labeled as fallow, litter w/supp. N, and inorganic-only control were measured in this study. Loads labeled inorganic-only 1984–1989 and 1977–1984 were measured in previous studies.

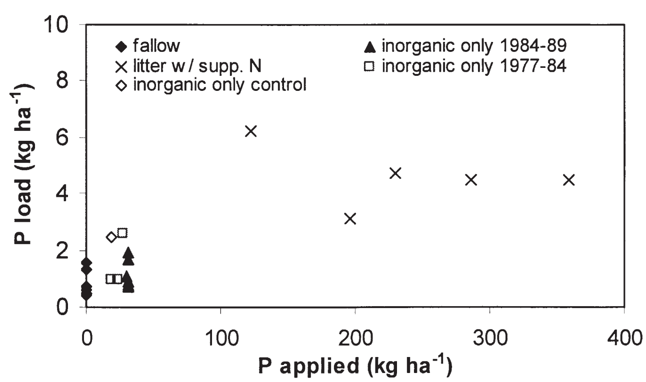


Fig. 9. Mean annual total P loads for the edge-of-field watersheds at Riesel. Loads labeled as fallow, litter w/supp. N, and inorganic-only control were measured in this study. Loads labeled inorganic-only 1984–1989 and 1977–1984 were measured in previous studies.

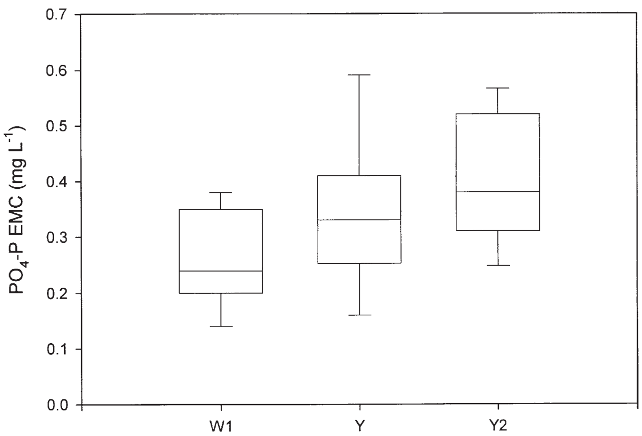


Fig. 10. Downstream event mean PO_4–P concentrations for the two litter application years. The P application rate increases from left to right.

tion, sediment filtration, and dilution also contribute to decreased loads and concentrations at the downstream watersheds. The spatial relationships between the litter application watersheds and the downstream watershed sites are shown in Fig. 1. For the downstream sites, 43 and 21% of the annual total N loss and 72 and 37% of the P loss was associated with eroded material in the two years following litter application (Tables 6 and 7). For the downstream watersheds, these annual percentages of sediment-bound nutrient loss were within the range of percentages from the edge-of-field watersheds. Although various land uses contribute considerable variability to downstream water quality, the percentages of sediment-bound nutrient loss were fairly similar to percentages from the cultivated watersheds, which contributed a majority of the nutrient load.

The reduced nutrient loss downstream emphasizes the danger of equating edge-of-field losses to watershed export or, in other words, the impacts of spatially distributed mixed land uses on downstream water quality. For the downstream sites, nutrient concentrations and loads tended to increase as nutrient application rate increased (on a per area basis on the entire watershed). This relationship is illustrated for PO_4–P concentrations (Fig. 10). Although this relationship between P loss and P application rate was fairly consistent in this study, the behavior and location of individual application fields within larger watersheds is also important in agricultural P management (Gburek and Sharpley 1998).

CONCLUSIONS

The water quality impacts resulting from the conversion from a traditional inorganic fertilizer strategy to a hybrid poultry litter and supplemental inorganic N strategy were evaluated. It should be kept in mind that the results represent new litter application fields with initially low and still relatively low soil P levels. Litter impacts on water quality for fields that have received numerous applications over a number of years would possibly be quite different.

Following litter application on the edge-of-field watersheds, concentrations of PO_4–P in runoff were positively correlated to litter application rate and to Mehlich-3 soil P levels; however, the relative contribution of P from each source is not well established and warrants further research. The increases in PO_4–P concentrations in each of the study years, which coincided with increases in Mehlich-3 soil P levels, are expected to continue if P application in excess of crop needs continues. In contrast to PO_4–P, NO_3–N and NH_4–N concentrations from cultivated watersheds in the second application year tended to decrease as the ratio of litter to inorganic fertilizer increased. Nutrient loads were not significantly related to litter rate due to differences in runoff volumes and were variable from year to year based on runoff characteristics. It is expected that P loads will increase, especially on high litter rate sites, as litter application continues.

At the three downstream sites, litter application did impact water quality but to a lesser extent than at the small edge-of-field watersheds. Thus, nutrient application to areas adjacent to water bodies should be avoided since mechanisms that tend to decrease nutrient loss are limited. Results also emphasize the error of equating edge-of-field losses to watershed export and downstream water quality impacts.

Although not the primary focus of this study, the impact of several management practices on resulting water quality was recognized. Inorganic fertilizer, in this case UAN, was more susceptible than poultry litter to loss in rains soon after application. The PO_4–P concentrations in runoff from the pasture watersheds with surface litter application tended to exceed those from cultivated watersheds with incorporated litter. The benefits of incorporation are offset to some degree by increased sediment-bound P loss from cultivated watersheds. Split application of N also tended to reduce NO_3–N concentrations in runoff. These initial results on new litter application fields support using the following management practices to reduce offsite nutrient transport: (i) apply fertilizer in split application or near the time when nutrients are needed by crops; (ii) avoid fertilizer application before heavy rainfall forecasts; (iii) apply only necessary crop nutrients; and (iv) recognize the differences in dissolved and sediment-bound nutrient loss based on land use type and application method.

Based on these results for new poultry litter application sites, an annual litter application rate of 2.2 to 4.5 Mg ha^{-1}, depending on litter nutrient content, is recommended. This recommended range of litter rates in conjunction with supplemental inorganic N application should provide the following benefits: (i) supply crop P and N requirements; (ii) enhance cropland and pasture soils with beneficial macro- and micronutrients and organic matter; (iii) at worst produce a slow increase in soil P levels; (iv) utilize the resource value of litter in a sustainable manner in contrast to disposal techniques; and (v) produce little or no negative water quality impact.

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REFERENCES

Bundy, L.G., T.W. Andraski, and J.M. Powell. 2001. Management practice effects on phosphorus losses in runoff in corn production systems. J. Environ. Qual. 30:1822–1828.

Chichester, F.W., and C.W. Richardson. 1992. Sediment and nutrient loss from clay soils as affected by tillage. J. Environ. Qual. 21:587–590.

Chinkuyu, A.J., R.S. Kanwar, J.C. Lorimor, and T.B. Bailey. 2002. Effects of laying hen manure application rate on water quality. Trans. ASAE 45:299–308.

Edwards, D.R., and T.C. Daniel. 1994. A comparison of runoff quality effects of organic and inorganic fertilizers applied to fescuegrass plots. Water Resour. Bull. 30:35–41.

Edwards, D.R., T.C. Daniel, J.F. Murdoch, and P.A. Moore, Jr. 1996. Quality of runoff from four northwest Arkansas pasture fields treated with organic and inorganic fertilizer. Trans. ASAE 39:1689–1696.

Edwards, D.R., P.A. Moore, Jr., T.C. Daniel, and P. Srivastava. 1995. Poultry litter-treated length effects on quality of runoff from fescue plots. Trans. ASAE 39:105–110.

Eghball, B., J.E. Gilley, D.D. Baltensperger, and J.M. Blumenthal. 2002. Long-term manure and fertilizer application effects on phosphorus and nitrogen in runoff. Trans. ASAE 45:687–694.

Gass, W.B. 1987. Plant soil and water testing laboratory recommendations. Texas Agric. Ext. Serv., College Station.

Gburek, W.J., and A.N. Sharpley. 1998. Hydrologic controls on phosphorus loss from upland agricultural watersheds. J. Environ. Qual. 27:267–277.

Gburek, W.J., A.N. Sharpley, L. Heathwaite, and G.J. Folmar. 2000. Phosphorus management at the watershed scale: A modification of the phosphorus index. J. Environ. Qual. 29:130–144.

Gilley, J.E., and L.M. Risse. 2000. Runoff and soil loss as affected by the application of manure. Trans. ASAE 43:1583–1588.

Green, W.R., and B.E. Haggard. 2001. Phosphorus and nitrogen concentrations and loads at the Illinois River south of Siloam Springs, Arkansas. Water Resour. Investigations Rep. 01-4217. USGS, Reston, VA.

Haan, C.T. 2002. Statistical methods in hydrology. 2nd ed. Iowa State Press, Ames.

Haggard, B.E., P.A. Moore, Jr., I. Chaubey, and E.H. Stanley. 2003. Nitrogen and phosphorus concentrations and export from an Ozark Plateau catchment in the United States. Biosyst. Eng. 86:75–85.

Hall, B.M. 1994. Broiler litter effects on crop production, soil properties, and water quality. M.S. thesis. Auburn Univ., Auburn, AL.

Harmel, R.D., K.W. King, and R.M. Slade. 2003. Automated storm water sampling on small watersheds. Appl. Eng. Agric. 19:667–674.

Helsel, D.R., and R.M. Hirsch. 1993. Statistical methods in water resources. Elsevier, New York.

Janzen, R.A., W.B. McGill, J.J. Leonard, and S.R. Jeffrey. 1999. Manure as a resource—Ecological and economic consideration in balance. Trans. ASAE 42:1261–1273.

Kleinman, P.J.A., R.B. Bryant, W.S. Reid, A.N. Sharpley, and D. Pimentel. 2000. Using soil phosphorus behavior to identify environmental thresholds. Soil Sci. 165:943–950.

Kleinman, P.J.A., and A.N. Sharpley. 2003. Effect of broadcast manure on runoff phosphorus concentrations over successive rainfall events. J. Environ. Qual. 32:1072–1081.

Kleinman, P.J.A., A.N. Sharpley, B.G. Moyer, and G.E. Elwinger. 2002. Effect of mineral and manure phosphorus sources on runoff phosphorus. J. Environ. Qual. 31:2026–2033.

Leidner, J. 2002. Fertilizer for free. Progressive Farmer, August.

Mehlich, A. 1984. Mehlich 3 soil test extractant: A modification of the Mehlich 2 extractant. Commun. Soil Sci. Plant Anal. 15:1409–1416.

Mengel, K., and E. Kirkby. 1978. Principles of plant nutrition. Int. Potash Inst., Worblaufen-Bern, Switzerland.

Minitab. 2000. MINITAB 12. Minitab, State College, PA.

Nichols, D.J., T.C. Daniel, and D.R. Edwards. 1994. Nutrient runoff from pasture after incorporation of poultry litter and inorganic fertilizer. Soil Sci. Soc. Am. J. 58:1224–1228.

Pierson, S.T., M.L. Cabrera, G.K. Evanylou, H.H. Kuykendall, C.S. Hoveland, M.A. McCann, and L.T. West. 2001. Phosphorus and ammonium concentrations in surface runoff from grasslands fertilized with broiler litter. J. Environ. Qual. 30:1784–1789.

Pionke, H.B., W.J. Gburek, R.R. Schnabel, A.N. Sharpley, and G.F. Elwinger. 1999. Seasonal flow, nutrient concentrations and loading patterns in stream flow draining an agricultural hill-land watershed. J. Hydrol. (Amsterdam) 220:62–73.

Ribaudo, M.O., N.R. Gollehon, and J. Agapoff. 2003. Land application of manure by animal feeding operations: Is more land needed? J. Soil Water Conserv. 58:30–38.

Sauer, T.J., T.C. Daniel, D.J. Nichols, C.P. West, P.A. Moore, and G.L. Wheeler. 2000. Runoff water quality from poultry litter treated pasture and forest sites. J. Environ. Qual. 29:515–521.

Sharpley, A.N., S.C. Chapra, R. Wedepohl, J.T. Sims, T.C. Daniel, and K.R. Reddy. 1994. Managing agricultural phosphorus for protection of surface waters: Issues and options. J. Environ. Qual. 23:437–451.

Sharpley, A.N., W.J. Gburek, G. Folmar, and H.B. Pionke. 1999. Sources of phosphorus exported from an agricultural watershed in Pennsylvania. Agric. Water Manage. 41:77–89.

Sharpley, A.N., and S.J. Smith. 1992. Water quality: Prediction of bioavailable phosphorus loss in agricultural runoff. J. Environ. Qual. 22:32–37.

Sharpley, A.N., S.J. Smith, and J.W. Naney. 1987. Environmental impact of agricultural nitrogen and phosphorus use. J. Agric. Food Chem. 35:812–817.

Technicon Industrial Systems. 1973a. Nitrate and nitrite in water and waste water. Industrial Method 100-70w. Bran-Luebbe, Roselle, IL.

Technicon Industrial Systems. 1973b. Ammonia in water and waste water. Industrial Method 98-70w. Bran-Luebbe, Roselle, IL.

Technicon Industrial Systems. 1976. Individual/simultaneous determination of nitrogen and/or phosphorus in BD acid digest. Industrial Method 344-74a. Bran-Luebbe, Roselle, IL.

Torbert, H.A., T.C. Daniel, J.L. Lemunyon, and R.M. Jones. 2002. Relationship of soil test phosphorus and sampling depth to runoff phosphorus in calcareous and noncalcareous soils. J. Environ. Qual. 31:1380–1387.

Torbert, H.A., K.N. Potter, and J.E. Morrison. 1996. Management effects on nitrogen and phosphorus losses in runoff on expansive clay soils. Trans. ASAE 39:161–166.

USDA and United States Department of Commerce. 1996. 1992 Census of Agriculture. Ranking of states and counties. AC92-S-3. Vol. 2. Subject Series Part 3. USDA, Washington, DC.

USDA and USEPA. 1999. Unified national strategy for animal feeding operations. USDA and USEPA, Washington, DC.

USDA National Agricultural Statistics Service. 1999. 1997 Census of Agriculture. Ranking of states and counties. AC97-S-2. Vol. 2. Subject Series Part 2. USDA, Washington, DC.

Vervoort, R.W., D.E. Radcliffe, M.L. Cabrera, and M. Latimore, Jr. 1998. Field-scale nitrogen and phosphorus losses from hayfields receiving fresh and composted broiler litter. J. Environ. Qual. 27:1246–1254.

Vories, E.D., T.A. Costello, and R.E. Glover. 2001. Runoff from cotton fields fertilized with poultry litter. Trans. ASAE 44:1495–1502.

Wood, B.H., C.W. Wood, K.H. Yoo, K.S. Yoon, and D.P. Delaney. 1999. Seasonal surface runoff losses of nutrients and metals from soils fertilized with broiler litter and commercial fertilizer. J. Environ. Qual. 28:1210–1218.